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Salt marshes for flood risk reduction: Quantifying long-term effectiveness and life-cycle costs

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ABSTRACT

Flood risks are increasing worldwide due to climate change and ongoing economic and demographic development in coastal areas. Salt marshes can function as vegetated foreshores that reduce wave loads on coastal structures such as dikes and dams, thereby mitigating current and future flood risk. This paper aims to quantify long-term (100 years) flood risk reduction by salt marshes. Dike-foreshore configurations are assessed by coupled calculations of wave energy dissipation over the foreshore, sediment accretion under sea level rise, the probability of dike failure, and life-cycle costs. Rising sea levels lead to higher storm waves, and increasing probabilities of dike failure by wave overtopping. This study shows that marsh elevation change due to sediment accretion mitigates the increase in wave height, thereby elongating the lifetime of a dike-foreshore system. Further, different human interventions on foreshores are assessed in this paper: realization of a vegetated foreshore via nourishment, addition of a detached earthen breakwater, addition of an unnaturally high zone, or foreshore build-up by application of brushwood dams that enhance sediment accretion. The performance of these strategies is compared to dike heightening for the physical boundary conditions at an exposed dike along the Dutch Wadden Sea. Cost-effectiveness depends on three main factors. First, wave energy dissipation, which is lower for salt marshes with a natural elevation in the intertidal zone, when compared to foreshores with a high zone or detached breakwater. Second, required costs for construction and maintenance. Continuous maintenance costs and delayed effects on flood risk make sheltering structures less attractive from a flood risk perspective. Third, economic value of the protected area, where foreshores are particularly cost-effective for low economic value. Concluding, life-cycle cost analysis demonstrates that, within certain limits, foreshore construction can be more cost-effective than dike heightening.

1. Introduction

Flood risks are rising in many coastal areas around the world due to a combination of climate change and ongoing intensification of human activities near coastlines. Current population density in low-elevation coastal zones is over five times higher than the global mean, and is expected to increase further in the coming decades (Neumann et al., 2015). Predictions indicate that economic development in coastal areas will result in a doubling of economic damage by storms like hurricanes, and that climate change will cause an additional doubling of this damage (Mendelsohn et al., 2012). Local impact of climate change can vary due to regional differences in sea level rise, climate change-induced variations in storm characteristics and associated extreme water levels (Wahl et al., 2017), and land subsidence due to sub-surface mining of oil, gas, and water (Syvitski et al., 2009).

These increasing flood risks encourage implementation of innovative solutions for coastal defense. In this context, efforts are being made to make greater use of nature-based approaches for flood risk reduction (Spalding et al., 2014). Coastal ecosystems, such as salt marshes, reefs and mangrove forests, contribute to flood risk reduction via wave energy dissipation (Möller et al., 2014) and shoreline stabilization (Gedan et al., 2011). These effects are sustained by adaptation to changing hydrodynamic conditions (Kirwan et al., 2016). Furthermore, coastal ecosystems provide additional ecosystem services such as nutrient cycling and detoxification, nursery ground for fish and oysters, and recreational activities (Barbier et al., 2011). In spite of the

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increasing scientific evidence for the flood risk reduction potential of coastal ecosystems, coastal defense is still not often the primary objective of salt marsh restoration projects (16%), in comparison to, for example, dune restoration studies (65%) (Morris et al., 2018).

In hybrid flood defenses, nature-based elements such as salt marshes are utilized as vegetated foreshores, which reduce hydraulic loads on the flood defense structure behind (Vuik et al., 2016). Wave energy dissipation is caused by a combination of wave breaking, bottom friction, and attenuation by vegetation (Möller et al., 1999). The strength of these dissipation mechanisms depends on the wave height to water depth ratio (Möller, 2006). In shallow coastal seas, waves are mostly depth-limited during severe storms, also during storms. Depth-limitation of waves makes foreshores highly effective. Salt marshes can retain their effectiveness under sea level rise (SLR) up to a certain rate, due to sedimentation (Kirwan and Megonigal, 2013) and sub-surface expansion by below-ground root growth (Nyman et al., 2006). Global measurements demonstrate that marshes are generally raising in elevation at rates similar to or exceeding historical SLR, and process-based models indicate that marshes are able to survive a wide range of future amplified SLR rates (Kirwan et al., 2016; Schuerch et al., 2018).

Costs are a crucial factor when selecting a certain measure for flood risk reduction. Since dikes with vegetated foreshores can adapt to SLR, proponents claim they are more sustainable and cost-effective than traditional engineering solutions, such as dikes and dams, in times of climate change (Temmerman et al., 2003; Sutton-Grier et al., 2015). According to Narayan et al. (2016), salt marshes and mangroves can be two to five times cheaper than submerged breakwaters for wave heights up to 0.5 m. Although this study made a great effort in collecting costs and benefits from many nature-based flood defenses, they only expressed benefits in terms of wave height reduction instead of in monetary value. Further, most dikes and dams are supposed to withstand wave heights substantially higher than 0.5 m, for which a comparison of cost-effectiveness is still missing. Reguero et al. (2018) have taken this a step further in a case study for the Gulf Coast of the United States, by comparing investments and averted damage between traditional engineering approaches and nature-based solutions. However, they only expressed the effectiveness of nature-based solutions in the form of general hazard reduction percentages found in the literature, which they assumed to be generally applicable. Therefore, there is still a need for concrete comparison between hard structures and hybrid flood defenses in terms of long-term effectiveness for flood risk reduction, and associated life-cycle costs.

This paper aims to quantify the long-term cost-effectiveness of salt marshes in reducing flood risk, in comparison to conventional dike heightening. Several possible human interventions on the foreshore are introduced, aiming to influence its biogeomorphological development and thereby the flood protection level and/or the expected lifetime of the hybrid flood defense. We express the performance of hybrid flood defenses in terms of reduction in probability of dike failure. Flood risk, which is the expected annual damage due to flooding, is generally defined as the product of probability and consequences of dike failure. Although foreshores affect multiple failure mechanisms, only dike failure due to wave overtopping is considered here, since (1) this is often one of the dominant mechanisms for coastal dikes (Danka and Zhang, 2015), (2) wave overtopping directly depends on dike crest level and SLR, and (3) it is affected by vegetated foreshores via wave height reduction (Vuik et al., 2016). The failure probability is calculated in a probabilistic assessment, in which the variabilities and uncertainties in storm, salt marsh, and dike characteristics are taken into account (Vuik et al., 2018b). Cost-effectiveness of different measures is assessed in a life-cycle cost analysis by comparing initial and future investment costs with averted damage (Vrijling, 2001).

This paper is organized as follows. First, three issues relevant for implementing vegetated foreshores for long-term flood risk reduction are discussed: stability during storms, long-term morphological development, and strategies for influencing flood risk reduction (section 2). This discussion is based on a combination of literature review and data analysis for evaluating morphological changes of salt marshes. Based on this discussion, the development in time of the failure probability of a dike with vegetated foreshore is computed, thereby taking into account SLR, vertical accretion in response to climate change, lateral foreshore dynamics, and the effect of several human interventions on foreshores (starting from section 4.1). Subsequently, the performance of different strategies is assessed and compared to traditional dike heightening, based on costs and benefits (starting from section 4.3). Finally, advantages and disadvantages of these strategies are discussed from a broader perspective, considering cost-effectiveness for flood risk reduction, required maintenance, environmental impact on the surrounding system, and provision of additional ecosystem services (section 5).

2. Foreshores in flood risk management

In this section, issues are discussed which are pertinent for implementing vegetated foreshores for long-term flood risk reduction: stability of salt marshes under storm conditions (section 2.1), long-term vertical and lateral dynamics of salt marshes (section 2.2), and strategies aiming to affect these dynamics and the associated effectiveness of the foreshore (section 2.3).

2.1. Stability of salt marshes during storms

To consider the effect of a salt marsh foreshore for flood risk reduction, the dike manager has to be convinced that the foreshore will be effective during extreme storm surges or hurricanes. Stability of salt marshes can be subdivided into morphological stability and vegetation stability. Morphological stability concerns the resistance against erosion and subsequent change in marsh elevation or marsh width. Vegetation stability is related to the strength of the vegetated top layer.

Concerning morphological stability, salt marshes are generally highly stable under wave forcing (Leonardi et al., 2016). For example, Gittman et al. (2014) reported absence of any change in marsh surface elevation due to a Category 1 hurricane with maximum wind speeds of 34 m/s, in contrast to significant damage and collapse of many bulkheads (vertical walls for shoreline protection) in the same area. Spencer

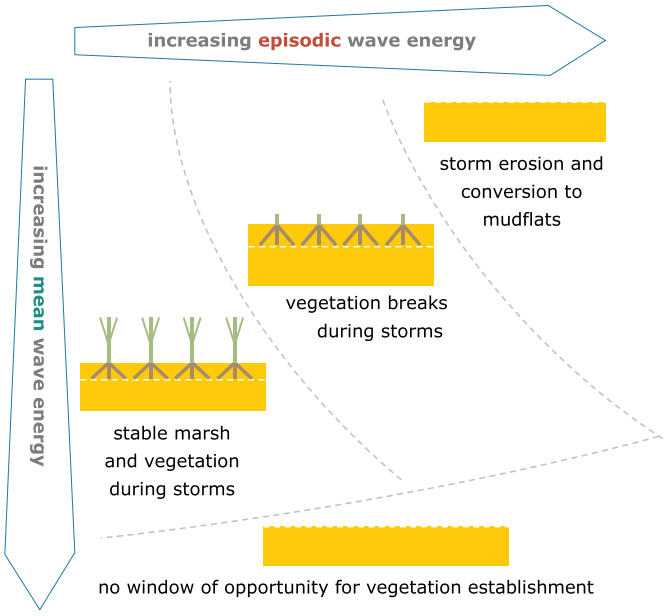
increasing episodic wave energy
increasing mean wave energy
storm erosion and conversion to mudflats
vegetation breaks during storms
stable marsh and vegetation during storms
no window of opportunity for vegetation establishment

Fig. 1. Conceptual view on marsh stability during storms, with higher vulnerability for relatively low mean wave energy, both for vegetation stability (middle) and marsh surface erosion (upper right).

et al. (2016) describe the impact of a sequence of simulated full-scale storm surge conditions in a wave flume, with only 6 mm average vertical lowering of the marsh surface. Storms can even cause net accretion on salt marshes (Turner et al., 2006).

However, other studies report extensive erosion of marshes during hurricanes (Morton and Barras, 2011), including large areas from which the marsh mat is torn away and immediately converted into open water (Cahoon, 2006). Marshes with such a substantial elevation loss commonly were either highly deteriorated and/or had a high organic matter content (Cahoon, 2006). According to Howes et al. (2010), differences in hurricane impact can be explained by soil composition and shear strength. Large-scale erosion only occurs in low-strength fresh and brackish marshes, with low day-to-day wave height and tidal amplitude. These marshes usually have a high organic content, and are sometimes affected by nutrients introduced via freshwater river diversions, which promotes poor rhizome and root growth (Kearney et al., 2011). Also sediment starvation can lead to increasing organic matter content, which results in structural weakness and edge failure (Peteet et al., 2018). Salt marshes rooted in mineral soils have much higher shear strengths, and are the most resilient wetlands to erosional storm impacts (Morton and Barras, 2011). Leonardi and Fagherazzi (2015) confirm that local variability in resistance of low-energy marshes might lead to unpredictable failure of large marsh portions with respect to average erosion rates. Marshes exposed to relatively high daily wave energy display constant and predictable average erosion rates, and low susceptibility to episodic severe storm events. In conclusion, marshes normally subject to significant wave energy are less susceptible to erosion during storms (Fig. 1), as long as windows of opportunity exist for plant seeds to germinate and seedlings to grow.

Second, stability of the vegetated top layer is important to maintain benefits from wave attenuation and bottom friction. Dense and tall vegetation is known to be highly effective in dissipating wave energy, both in emerged (e.g., Anderson et al. (2014)) and submerged conditions (e.g., Vuik et al. (2016)). However, stem breakage may occur as wave height increase, thereby limiting the energy dissipating factors to bottom friction on a rough salt marsh surface with the remnants of vegetation (Liffen et al., 2013; Silinski et al., 2015). The maximum wave force that a plant stem can withstand depends on its mechanical properties, such as stem height and diameter, flexibility, and frontal area (Vuik et al., 2018a). For high waves (significant wave heights over 1 m), relatively tall plant species will lose the majority of their aboveground biomass; a large-scale flume experiment showed 80% stem breakage of *Elymus athericus* (Rupprecht et al., 2017), and 50% reduction in stem density of *Spartina alterniflora* was observed after a Category 1 hurricane (Gittman et al., 2014). Plants at locations exposed to higher mean wave energy develop shorter and thicker stems, which makes them less vulnerable to stem breakage (Silinski et al., 2018). This implies that, similar to morphological stability, locations with low mean wave energy are most sensitive to stem breakage during severe episodic storm events (Fig. 1).

2.2. Temporal development of salt marshes

For taking into account the effect of a salt marsh foreshore in the design of a dike construction or reinforcement, a life-cycle analysis is required over the expected life time of the structure. This analysis takes into account SLR, as well as the response of marsh width and surface elevation to SLR.

Without adaptation, salt marshes will begin to disappear due to increased rates of SLR (Craft et al., 2009), as more frequent submersion will drown the vegetation and convert what was previously vegetated marsh to open water or bare mudflats (Kirwan and Megonigal, 2013). However, marshes can change in elevation due to biogeomorphological interactions such as sediment accretion (Temmerman et al., 2003) and sub-surface expansion due to root growth (Nyman et al., 2006). These processes result in marsh building at rates similar to or exceeding historical SLR (Kirwan et al., 2016). Marsh survival is expected to depend primarily on submergence time and suspended sediment concentrations (Kirwan et al., 2010; D'Alpaos et al., 2011), which explains the rapid loss of Jamaica Bay's salt marshes (Hartig et al., 2002). Recent simulations suggest that the resilience of wetlands is primarily driven by the accommodation space available for accumulation of fine sediments. Schuerch et al. (2018) state that coastal managers can strongly influence the survival of salt marshes under climate change, since accommodation space is mainly constrained by built infrastructure in the coastal zone. Tidal wetlands can be safeguarded by facilitating their landward migration under SLR, for example via inland displacement of hard coastal structures.

In addition to vertical variations, marshes can also display lateral dynamics due to seedling establishment and subsequent expansion, and

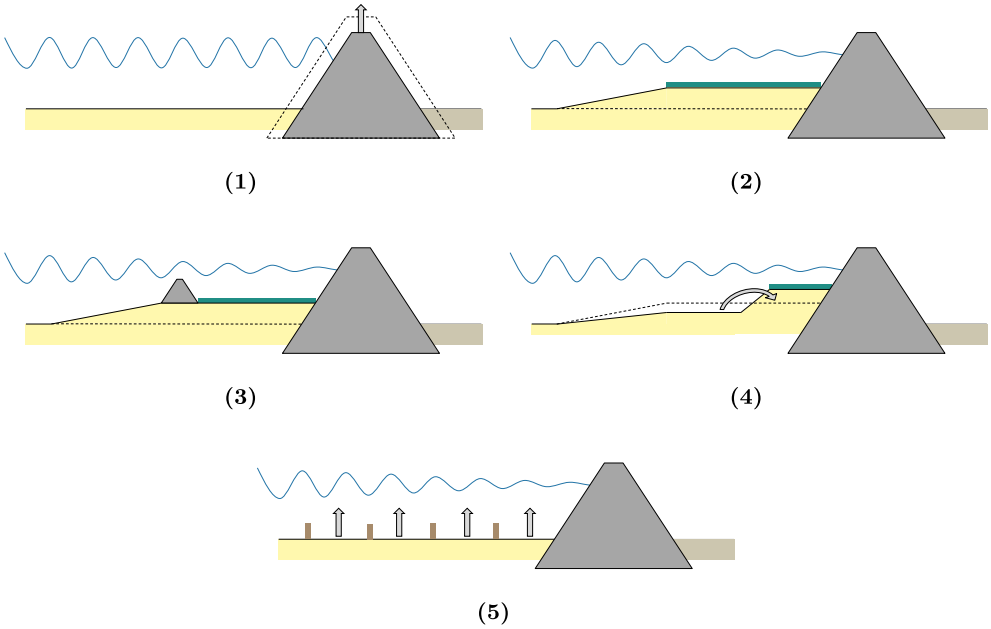

Fig. 2. Strategies for reducing flood risk, via dike heightening, or via human interventions on the foreshore. (1) dike heightening, (2) salt marsh construction, (3) salt marsh with detached structure, (4) marsh rejuvenation and high zone, (5) brushwood dams and sediment accretion.

lateral erosion of the marsh edge (Van der Wal et al., 2008). The marsh edge can shift several meters per year and shows cyclic alternations between erosion and expansion on decadal or longer timescales (Allen, 2000; Singh Chauhan, 2009). These variations in marsh width are driven by the interplay between sediment dynamics at the bare mudflat and within the salt marsh (Balke et al., 2016; Bouma et al., 2016).

This paper addresses the importance of vertical and lateral dynamics for failure probabilities, see section 3.2 and Appendix A.

2.3. Strategies for influencing flood risk reduction

Application of any kind of nature-based solution should fit in the surrounding physical system (De Vriend et al., 2015), and requires an analysis of technical feasibility, legal framework, integration in the landscape, and long-term influence on ecosystem services (Borsje et al., 2011). Generally speaking, vegetated foreshores can be realized or restored in front of a dike via human interventions, both for immediate flood risk reduction and in anticipation of future higher flood risk due to climate change. Here, we summarize several strategies considered in this paper, including strategies that involve human interventions on the foreshore (Fig. 2).

1. The most common strategy in response to SLR is traditional dike heightening. However, dike heightening can be difficult, for example, in cases where buildings are close to the dike or when the subsoil is too soft to support a heavy dike.
2. The most straightforward engineering approach to realizing a vegetated foreshore is via sediment nourishment. After nourishing the original bottom, the core material is covered by a top layer of clayey silt, on which salt-tolerant salt marsh species can germinate, emulating a natural salt marsh. Costs depend on water depth, required elevation, construction method, and availability of local sediment. Use of dredged sediments from shipping channels and harbors can be considered here. High mean wave energy and nutrient-poor sediment diminish the chance of successful vegetation establishment on such a constructed foreshore (Penning et al., 2016). Therefore, salt marsh realization is only feasible in relatively sheltered coastal systems. For sea dikes with severe waves, a sandy foreshore without vegetation may be more realistic than a vegetated foreshore (Oosterlo et al., 2018).
3. Salt marsh formation or restoration may also be enabled by detached structures, such as breakwaters or stone sills (Currin et al., 2008), which traps sediments from tidal flow to enable ongoing marsh accretion. Such hybrid combinations of natural and engineered shorelines are sometimes referred to as ‘living shorelines’ (Davis et al., 2015) or ‘green infrastructure’ (Silva et al., 2017). Presence of structures can lead to higher marsh elevations (Gittman et al., 2014), enhanced erosion resistance during storms, and higher stem densities (Smith et al., 2018), compared to natural marshes. During storms, both the detached structure and the salt marsh can help in reducing wave loads on the flood defense behind the living shoreline.
4. In many places, mature high salt marshes are present with generally lower species richness and nature value than lower, younger, and more dynamic pioneer marsh zones (Bakker et al., 2002). Here, safety and nature goals may be combined, by lowering and thereby rejuvenating the seaward part of the marshes. Dredged sediments can be used to construct a wave breaking high zone on the foreshore, directly in front of the dike. Since this zone is well above the level where natural accretion can be effective, newly accumulated sediments should be moved from the lower, natural part of the foreshore to the high zone, to preserve its wave damping functioning.
5. At exposed coasts, strong waves and currents may impede settling of fine sediments and establishment of salt marsh vegetation or mangroves (Winterwerp et al., 2013). Construction of a system with brushwood dams (Fig. 3) creates shelter, facilitates sedimentation, and prevents lateral erosion. Combining these dams with drainage ditches improves consolidation and aeration of the settled sediment. This method (known as ‘salt marsh works’) has successfully been applied for centuries, and has led to artificial salt marshes along 450 km of the Dutch, German and Danish Wadden Sea coastline (Bakker et al., 2002; Hofstede, 2003).

3. Methods

This section describes the methods for assessing long-term effects of vegetated foreshores on flood risk. Long-term development of foreshores is affected by marsh accretion induced by SLR (Section 3.2), lateral marsh dynamics (Appendix A.1) and human interventions (3.4). Finally, the methods used to assess costs (3.5) and cost-effectiveness are described (3.6).

Fig. 3. Brushwood dams in the Dutch Wadden Sea, which create artificial shelter for salt marsh vegetation from waves and currents. The dike is visible in the background (Photo: V. Vuik).

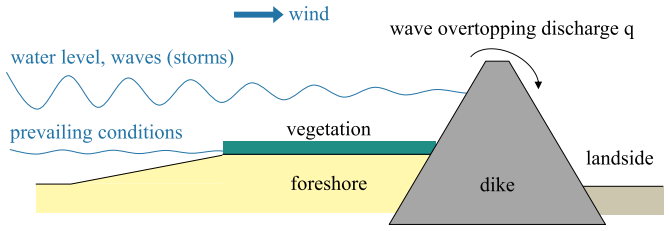

Fig. 4. Schematic representation of a dike-foreshore system (vertical dimensions not to scale).

3.1. System description and probabilistic modeling approach

Sea level rise, salt marsh dynamics, and human interventions affect the failure probability of a hybrid flood defense, which consists of a dike and a salt marsh foreshore (Fig. 4).

The probabilistic model of Vuik et al. (2018b) is used for evaluating the failure probability of the hybrid flood defense for different points in time under SLR (see section 3.2). The main characteristics of this model are summarized here for clarity. A wave overtopping discharge q is computed using EurOtop (2016), including uncertainty in the associated model parameters. This wave overtopping discharge is compared with a tolerable wave overtopping discharge q_{max}, which depends on the erosion resistance of the dike crest and rear slope. Dike failure is described via a Limit State Function $Z = q_{max} - q$, which becomes negative when the wave overtopping discharge exceeds the maximum tolerable value. The probability of failure P_f is equal to the probability $P(Z < 0)$.

Comparison of failure probabilities for different foreshore configurations requires a relatively quick and simple calculation method. Therefore, wave propagation over the vegetated foreshore is computed by means of a one-dimensional wave energy balance:

$$\frac{dEc_g}{dx} = S_{in} - S_{ds,w} - S_{ds,b} - S_{ds,f}, \tag{1}$$

where $E = (1/8)\rho g H_{rms}^2$ is wave energy density (J/m^2), H_{rms} root mean square wave height (m), ρ density of water (kg/m^3), g gravitational acceleration (m/s^2), c_g group velocity (m/s), and x distance (m) along the foreshore. Source terms (J m^{-2} s^{-1}) affect the amount of wave energy: input by wind (S_{in}), and dissipation via whitecapping ($S_{ds,w}$), depth-induced wave breaking ($S_{ds,b}$) and bottom friction ($S_{ds,f}$). The energy balance contains the wave breaking model of Battjes and Janssen (1978) with breaker parameter γ following Battjes and Stive (1985), with a standard deviation of 0.05. To avoid an overestimation of the wave height reduction for relatively long foreshores, the processes wind input (due to Snyder et al. (1981)) and whitecapping (due to Komen et al. (1984)) are activated. The energy balance is discretized, using a first order numerical scheme with step size $\Delta x = 10$ m. The offshore wave period T_p is considered in the energy balance, and the mean wave period $T_{m-1,0}$ in the overtopping calculations. A change in mean wave period $T_{m-1,0}$ on the foreshore is computed, using the equation of Hofland et al. (2017), with multiplication factor f_T with mean value of 1.00 and standard deviation of 0.09.

Most salt marsh vegetation will break under design conditions at exposed salt marshes (Vuik et al., 2018b). Therefore, the modeling approach with wave energy dissipation by cylindrical elements, combined with a calculated fraction of broken stems per grid cell, is replaced by a much simpler representation by means of bottom friction according to Madsen et al. (1988). A Nikuradse roughness height $k_N = 0.05 \pm 0.02$ m is used, based on Manning values typically used for marshland in hurricane conditions (Wamsley et al., 2010). For bare tidal flats, $k_N = 0.001$ m is applied. The resulting simplified model framework is shown in Fig. 5.

Boundary conditions for this model are a uniform wind speed and water level, and a wave height and period at the offshore boundary at the tidal flats, which are all specified via extreme value distributions with Gaussian correlation between parameters. For wave propagation over foreshores with submerged breakwaters, the formula of Van der Meer et al. (2005) for wave transmission over smooth, low-crested structures is added to the wave energy balance. The dike geometry, still water level, and wave conditions at the landward end of the foreshore are used for calculating a wave overtopping discharge.

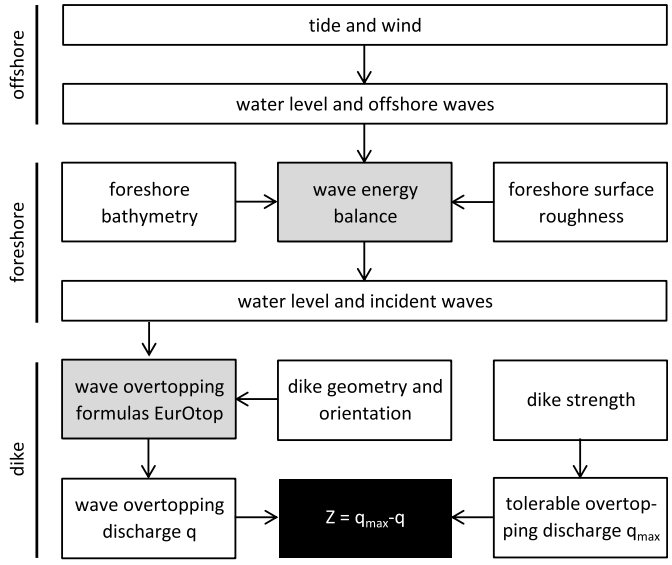

Fig. 5. Model framework to compute a probability of failure. The Limit State Function Z is defined as the difference between strength (tolerable overtopping discharge) and load (actual wave overtopping discharge).

The failure probability is computed using the probabilistic method FORM (Hasofer and Lind, 1974). This method iteratively draws numbers from all probability distributions and assesses the response of the Limit State Function via the model framework (Fig. 5). The end result is a probability of failure and the most likely combination of parameter values that leads to failure (the so-called design point). Table 1 shows how the failure probability declines for increasing dike crest level. Further, the return period $T = 1/P_f$ is included, which can be interpreted as the average number of years between consecutive dike failures.

3.2. Sea level rise and marsh elevation change

Time-dependent scenarios for SLR are adopted from Table AII.7.7 of IPCC (2013). The mean values of the IPCC scenario with the highest sea level change in 2100 are used here: RCP8.5 with 0.74 m in 2100, compared to the reference period 1986–2005 ($t = 0$ in the central year 1996). Waves in the shallow Wadden Sea are depth-limited, implying that wave heights will increase as sea level rises (Arns et al., 2017). This

Table 1
Annual failure probability P_f and return period T for dikes with different crest levels z_c and a foreshore at 0 m above mean sea level (MSL), including the boundary conditions in the design point: wind speed U_{10}, still water level ζ, offshore significant wave height H_{m0} and offshore mean wave period $T_{m-1,0}$.

z_c (m MSL)	P_f (−)	T (year)	U_{10} (m/s)	ζ (m MSL)	H_{m0} (m)	$T_{m-1,0}$ (s)
4	$8.1{\cdot}10^{-2}$	12	22.2	3.29	1.03	4.27
5	$8.6{\cdot}10^{-3}$	120	25.2	3.92	1.34	4.80
6	$5.4{\cdot}10^{-4}$	1900	27.9	4.55	1.62	5.22
7	$1.8{\cdot}10^{-5}$	56,000	30.3	5.14	1.89	5.58
8	$2.4{\cdot}10^{-7}$	$4.1{\cdot}10^{6}$	32.3	5.66	2.16	5.91
9	$1.1{\cdot}10^{-9}$	$9.5{\cdot}10^{8}$	33.8	6.08	2.40	6.19
10	$1.3{\cdot}10^{-12}$	$7.7{\cdot}10^{11}$	34.9	6.39	2.64	6.45

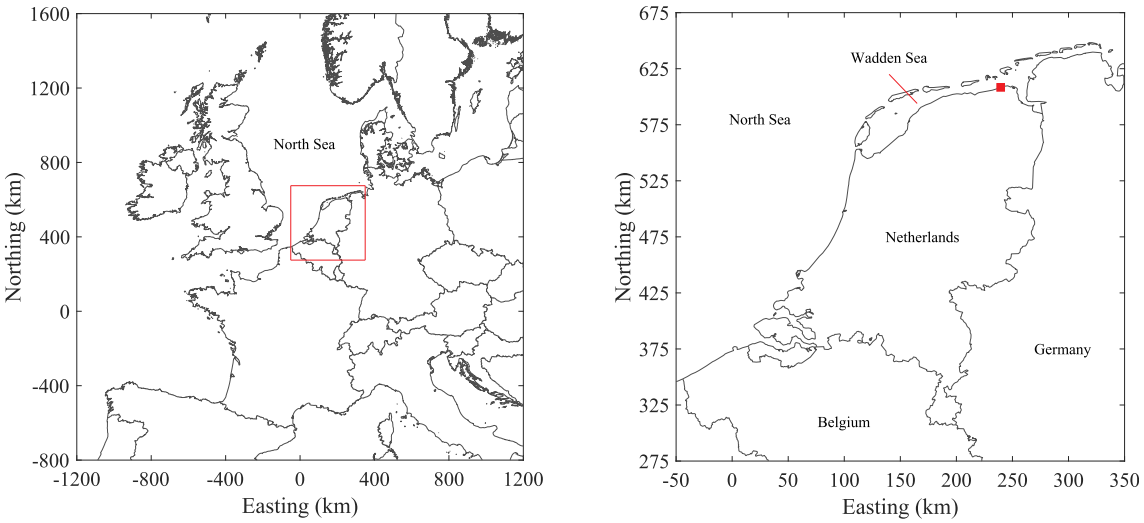

Fig. 6. Location of the Wadden Sea in Europe (left) and location of the salt marshes along a dike in the Netherlands (right panel, red square). Taken from Vuik et al. (2018b). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

effect is taken into account by keeping the wave height to water depth ratio and wave steepness equal to the situation without SLR.

For changes in marsh elevation, two basic situations are considered first: no temporal change at all ($dz_{fs}/dt = 0$), and a rate of change exactly equal to the rate of SLR ($dz_{fs}/dt = R(t)$), where $R(t)$ is the time-varying rate of SLR. In addition, a rate of change is estimated using the analytical approach by D'Alpaos et al. (2011). They describe how the accretion rate depends on the marsh elevation within the tidal range and suspended sediment concentration (SSC). The accretion rate decreases linearly from k at MSL to zero at Mean High Water H, according to the following differential equation proposed by D'Alpaos et al. (2011):

$$\frac{dz}{dt} = k\left(1 - \frac{z}{H}\right) - R(t), \tag{2}$$

in which z is the marsh elevation relative to a changing mean sea level, such that $dz/dt = dz_{fs}/dt - R(t)$, where z_{fs} is the foreshore elevation relative to MSL at $t = 0$. IPCC scenario RCP8.5 can be approximated by an accelerating sea level change $R(t) = R_0 + at$, with $R_0 = 2.14$ mm/year in 1996 and $a = 0.095$ mm/year^2. For an arbitrary initial foreshore elevation $z_{fs,0}$, a time-varying elevation $z_{fs}(t)$ is found using the following solution of the differential equation, Eq. (2):

$$z_{fs}(t) = c_1 H + (z_{fs,0} - c_1 H)e^{-kt/H} - \frac{aH}{k}t, \tag{3}$$

in which the dimensionless constant c_1 reads

$$c_1 = 1 + \frac{aH}{k^2} - \frac{R_0}{k} \approx 1. \tag{4}$$

3.3. Case study characteristics

The development of this failure probability over time is investigated in this paper for a case study located in the Dutch Wadden Sea. The Wadden Sea is a shallow coastal sea, separated from the North Sea by a series of barrier islands, stretching for over 500 km along the coasts of Denmark, Germany, and the Netherlands. It consists entirely of a sandy-muddy system of interconnected channels, tidal flats, and salt marshes. Barrier islands are separating the Wadden Sea from the North Sea. The tidal flats and salt marshes form the largest coherent habitat of this type in Europe and are together an essential element of the Wadden Sea ecosystem, which is a biologically highly productive ecosystem of great natural, scientific, economic and social importance. Because of its unique size and ecological value, large parts of the Wadden Sea have been designated as UNESCO World Heritage Sites since 2009. Denmark, Germany, and the Netherlands have adopted the Trilateral Wadden Sea Plan (2010), which states that 'sustainable human use will continue and have to be continuously balanced in a harmonious relationship between the needs of society and ecological integrity' (CWSS, 2017). Future flood risk reduction strategies should be evaluated in that context.

Within the Wadden Sea, a case study location has been selected, situated along the dike that protects the province Groningen in the Netherlands (Fig. 6). Man-made salt marshes are present here along approximately 25 km of the dike between Eemshaven and Lauwersmeer. The same case study location is used in Vuik et al. (2018b), where a detailed description of all site parameters can be found. Here, we only summarize the main characteristics of the dike and the existing salt marsh (stochastic parameters are presented as mean value ± standard deviation).

The dike has a crest level z_c at 8 m above MSL, and an outer slope angle $\tan(\alpha_d) = 1/4$. The tolerable overtopping discharge $q_{max} = 63 \pm 19$ l s^{-1} m^{-1}. For the actual dike height of 8 m MSL, no measure is needed, and the 'do nothing' strategy is economically most attractive. In this paper, we would like to obtain a situation with a need for flood risk reduction via either traditional dike heightening or a nature-based solution with foreshores. Therefore, the dike height is lowered to 6 m MSL in all calculations, for which flood risk reduction should be considered.

The foreshore is simplified, with a flat vegetated part with a width B_{fs} of 300 ± 50 m, an elevation z_{fs} of 1.5 ± 0.2 m MSL (close to Mean High Water), and a 1:100 slope from the marsh edge to the adjacent tidal flats at 0 m MSL (Fig. 4). The standard deviations reflect spatial variations, measurement inaccuracy, and possible erosion under design conditions. Vegetation is applied above 0.6 m MSL (including SLR), which is the actual marsh edge position in the Wadden Sea (Dijkema et al., 1990).

For Wadden Sea conditions, with a tidal amplitude H of approximately 1.5 m and a mean SSC of 70 mg/L (Borsje et al., 2008), a maximum vertical accretion rate of $k = 54$ mm/year is given in D'Alpaos et al. (2011). For these numbers, Eq. (2) results in 0.48 m gross accretion in 2100, which is less than 0.74 m SLR in the same year (Fig. 7).

Table 2 summarizes characteristics of the boundary conditions for this location.

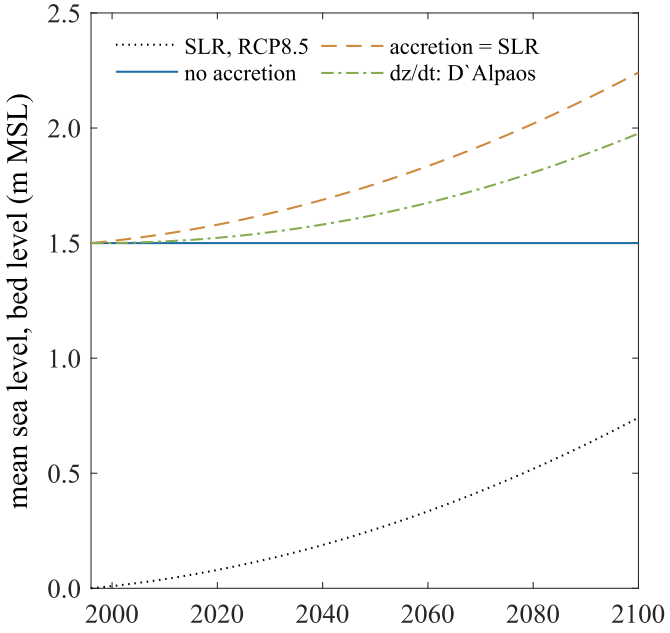

Fig. 7. Sea level rise (SLR) according to RCP8.5, and corresponding scenario's for marsh accretion: no accretion, accretion equal to SLR, and accretion according to Eq. (3).

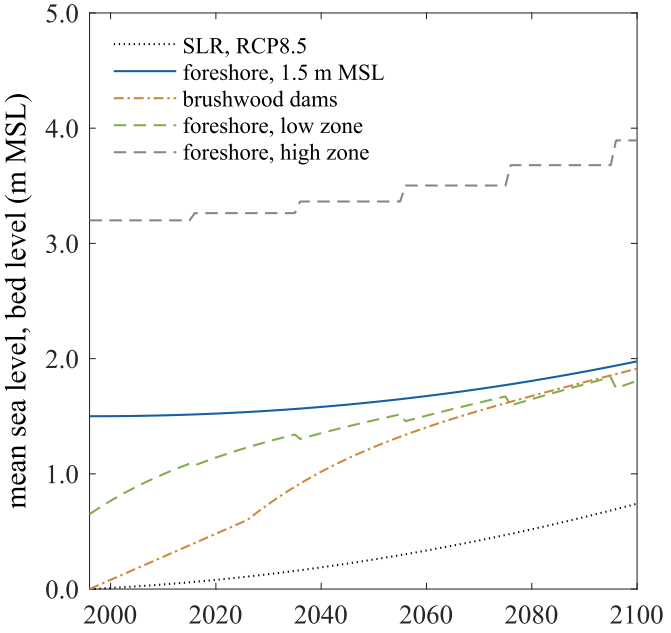

Fig. 8. Sea level rise (SLR) according to RCP8.5, and elevation in time for a nourished foreshore (1.50 m MSL at $t = 0$), brushwood dams, and a foreshore consisting of a low zone (0.65 m MSL at $t = 0$), and a high zone (3.20 m MSL at $t = 0$) which is regularly heightened.

Table 2
Probability distributions, characteristic values and Pearson's correlation coefficients for boundary conditions.

Parameter	Wind speed	Still water level	Significant wave height	Mean wave period
Symbol	U_{10}	ζ	H_{m0}	$T_{m-1,0}$
Units	m/s	m MSL	m	s
Distribution	GEV	GEV	Weibull	Weibull
Shape parameter	−0.14	−0.08	2.33	5.33
Scale parameter	2.70	0.38	0.70	3.62
Location parameter	17.5	2.45		
1/10 year value	22.7	3.23	1.00	4.23
1/100 year value	26.7	3.91	1.35	4.82
1/1000 year value	29.5	4.47	1.60	5.20
Correlations ρ:				
with ζ	0.80			
with H_{m0}	0.91	0.96		
with $T_{m-1,0}$	0.88	0.97	0.99	

3.4. Characteristics of strategies

Different strategies for flood risk reduction are considered, as introduced in section 2.3. This section describes how these strategies are technically accounted for in the failure probability calculations and cost estimates. All foreshores between the level above which vegetation establishes (0.6 m MSL) and Mean High Water (1.5 m MSL) are assumed to display an accretion rate according to Eq. (2), see Fig. 8.

1. First, we consider traditional dike heightening from 6 to 7 m MSL.
2. Next, we consider construction of a 300 m wide foreshore at 1.5 m MSL via nourishment, with standard deviations of 50 m and 0.2 m, respectively, to account for spatial variations. The bathymetry of this foreshore resembles that of a natural mature salt marsh. Vegetation is expected to establish, and accretion occurs in tandem with SLR (Fig. 8). High foreshores are most efficient from an engineering perspective, whereas low foreshores may be more suitable for establishment of ecologically valuable salt-tolerant plant species. The foreshore elevation can be used for optimizing the performance of a design. For the computation, we only consider the value of 1.5 m MSL.
3. A possible design alternative is the addition of an earthen or rubble mound breakwater at the marsh edge of the same foreshore as (2.). A small detached earthen breakwater is considered, with a 3 m wide crest at 2.5 m MSL (i.e., 1.0 m above ground level of 1.5 m MSL) and 1:6 slopes. We neglect possible influence of the breakwater on accretion rates. The standard deviation for marsh width is set to zero in this simulation.
4. The constructed foreshore can also be designed with a geometry deviating from that of a natural salt marsh. In this paper, we propose a foreshore with an identical volume to (2.), but consisting of a 100 m wide high zone at 3.20 m MSL directly in front of the dike (which is several wave lengths long, sufficient for wave breaking), bordered by a 200 m wide lower zone at 0.65 m MSL. This lower zone is higher than the level of 0.60 m MSL for which pioneer vegetation establishes (section 3.3). The slope angle between both zones is 1:25, which is expected to be stable during storms. Every 20 years, newly accumulated sediments on the low zone are displaced to the high zone via earthmoving, in such way that the high zone keeps pace exactly with SLR (Fig. 8, two dashed lines). The initial elevation of the high zone is iteratively chosen, so that its initial effect (in 1996) is identical to that of 1 m dike heightening, due to intense wave breaking on the high zone.
5. Finally, we consider facilitation of accretion by means of brushwood dams, resulting in an initial constant accretion rate of 2 cm/year on the bare tidal flats, as typically found in salt marsh works in the Wadden Sea (Hofstede, 2003). When the bed level exceeds the threshold of 0.6 m MSL (MSL after SLR), vegetation starts to settle, and accretion follows Eq. (2).

3.5. Costs of strategies

Construction and maintenance costs are estimated for the different strategies of Fig. 2, and included in Table 3. All unit prices are converted to 2018 price level, based on historic values of the consumer price index in the Netherlands, according to the Central Office for Statistics (CBS). Net Present Value (NPV) is calculated, using an interest

Table 3
Cost comparison of the different strategies from Fig. 2. *The mentioned costs for the detached earthen breakwater and earthmoving to high zone are in addition to the costs for foreshore construction, which is also present in these strategies.

	1	2	3	4	5
	Dike heightening, 1 m	Foreshore construction	Detached earthen breakwater*	Earthmoving to high zone*	Brushwood dams Wadden Sea
Unit price construction	4.5–12.4 M€	2.3–6.7 €	7.5–12.5 €	0.4 €	130 €
Unit price maintenance	x	x	x	3.0–10.0 €	22 €
Maintenance interval	x	x	x	20 years	1 year
Unit	km	m^3	m^3	m^3	m^1
Source	*1	*1	*2	*3	*4
Year of unit prices	2006	2013	1995	1995	2018
Unit price constr., 2018	5.4–14.9 M€	2.4–7.0 €	11.5–19.2 €	0.4 €	100 €
Unit price maint., 2018	x	x	x	4.6–15.4 €	22 €
Units/km, constr	1	525,000	20,500	525,000	2200–5000
Units/km, maint	x	x	x	6000–21,000	2200–5000
Constr. costs/km (M€)	5.4–14.9	1.3–3.7	0.20–0.40	0.20	0.50
Maint. costs/km/interval	x	x	x	0.03–0.32	0.05–0.11
NPV 1996–2100 (M€)	5.4–14.9	1.3–3.7	0.20–0.40	0.25–0.37	1.9–3.6

Sources for unit costs:
*1 = Jonkman et al. (2013).
*2 = https://www.bodemrichtlijn.nl/Bibliotheek/bodemsaneringstechnieken/c-grondverzet/c6-herstellenbodemprofiel/herstellen-bodemprofiel-kosten, visited 15-6-2018.
*3 = https://www.bodemrichtlijn.nl/Bibliotheek/bodemsaneringstechnieken/c-grondverzet/c2-open-ontgraven/openontgraven-kosten-verwijderen-grond, visited 15-6-2018.
*4 = Personal Communication Rijkswaterstaat, division Northern Netherlands.

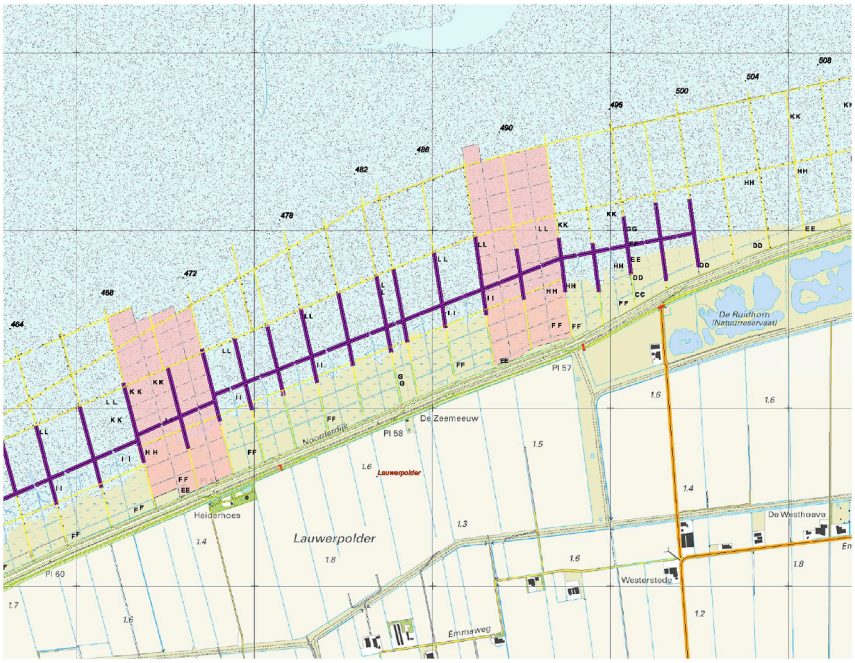

Fig. 9. Salt marsh works at the case study location in the Dutch Wadden Sea, with the dike (green line) salt marshes (green shading at the north of the dike), monitored sections (pink shading), maintained brushwood dams (purple lines) and abandoned brushwood dams (yellow lines). Source: Rijkswaterstaat Northern Netherlands. Distance between grid lines is 1 km. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

rate of 3% per year over the period 1996–2100. Costs for maintenance of the dike itself are not taken into account, assuming that dike heightening or interventions on the foreshore do not affect the required maintenance for the dike.

1. Costs for 1 m heightening of 1 km dike are based on numbers for rural area in Jonkman et al. (2013).
2. Costs for foreshore construction (Fig. 2, configuration 2) are based on transforming tidal flats (0 m MSL) into 300 m wide marshland (1.5 m MSL) via dredging and nourishing, with a unit price of 2.4–7.0 €/m^3. The resulting costs of 1.3–3.7 M€ per km are equivalent to 3.6–10.4 €/m^2 marshland. Costs are relatively high, compared to the average costs of 0.9 €/m^2 (range 0.01–28 €/m^2) in Narayan et al. (2016) and 3.5 €/m^2 in Jonkman et al. (2013) (price level 2007 converted to 2018), due to the substantial raising by 1.5 m. It is assumed that no significant maintenance is needed to preserve the foreshore.
3. Costs for a detached earthen low-crested breakwater (Fig. 2, configuration 3) are based on unit prices of clay for construction purposes, for a 1 m high dam on existing marshes, with slope angles of 1:6 and a crest width of 3 m. These costs are in addition to the costs for constructing the foreshore (configuration 2), which is also present in this alternative. It is assumed that maintenance costs for the earthen breakwater are negligible.

4. For constructing the foreshore with a high and low zone (Fig. 2, configuration 4), the volume is identical to that of a uniform foreshore at 1.5 m MSL. We assume that costs for dredging and nourishing are equal for both foreshore geometries, except for a surcharge of 2 €/m^2 for more accurate earthworks profiling on the high zone. This equates to 0.2 M€ per km dike, equivalent to an additional 0.4 € per m^3 foreshore volume. For periodic earthmoving on existing marsh, unit costs are based on earthmoving of in-situ sediments, using land equipment in wet circumstances (4.6–15.4 €/m^3). All these costs are in addition to the costs of constructing the foreshore (configuration 2), which is also present in this alternative.
5. For salt marsh works with brushwood dams (Fig. 2, strategy 5), the contribution of maintenance to the overall costs will be substantial. Costs are based on two zones with sedimentation fields of 300 m wide (perpendicular to the dike) and 200 m long (parallel to the dike), based on the Dutch Wadden Sea (Fig. 9). The sedimentation fields at the seaside will lead to raising of the tidal flats, while salt marshes are supposed to emerge in the sedimentation fields at the landside. For constructing this system, a 5 km brushwood dam is needed per 1 km dike. For a stable, mature salt marsh system, only dams near the marsh edge have to be maintained. Currently, 111 km of brushwood dams are maintained for approximately 50 km of coastline with salt marshes (Fig. 9, and personal communication with salt marsh manager at Rijkswaterstaat), which is 2.2 km of dam per 1 km of dike. Consequently, maintenance costs will become lower after several decades. Average maintenance costs in the Dutch Wadden Sea are equivalent to 22 €/m per year. In Indonesia, similar dams are applied for mangrove restoration. For comparison: a combination of bamboo poles and brushwoods is relatively expensive (170 €/m for construction, 80 €/m per year for maintenance, personal communication Witteveen + Bos, Indonesia, and Wilms et al. (2018)). Cheaper and more durable materials are currently being tested, such as horizontal bamboo beams between the poles (45 €/m for construction, 15 €/m per year for maintenance).

3.6. Cost-effectiveness of strategies

In this paper, we determine which strategy is most cost-effective, depending on the damage in case of dike breaching. We only consider benefits in terms of flood risk reduction, the primary objective of flood risk reduction projects, either traditional or nature-based. Although relevant for decision making, economic value of other ecosystem services is not quantified in this study. The most cost-effective strategy leads to the minimum sum of investment costs and expected damage. Investment costs I consist of initial construction costs and the NPV of maintenance costs. Expected damage is defined as the product of time-varying annual failure probability $P_f(t)$ and damage D in case of dike breaching: the expected value of the annual damage.

Total costs C_{tot} are defined as

$$C_{tot} = \sum_t \frac{I}{(1+r)^t} + \sum_t \frac{P_f(t)D}{(1+r)^t}, \tag{5}$$

in which r is the interest rate (3%) and t the time since $t = 0$ (1996). For this assessment, the mean values of all cost ranges in Table 3 are used. Total costs depend on the damage in the area that is being protected. The optimal strategy will thus also be influenced by this damage.

4. Results

This section presents computed failure probabilities of dike-foreshore configurations, affected by SLR and vertical accretion (section 4.1), lateral marsh edge dynamics (4.2) and human interventions on the foreshore (4.3). The influence on failure probabilities is evaluated in relation to the associated costs (4.4).

4.1. Sea level rise and accretion rates

The development of the annual failure probability P_f in time is investigated for a dike with existing foreshore, under different accretion rates (Fig. 7). The failure probability increases in time, since SLR leads to a decline in freeboard (i.e., crest level minus still water level) of the dike (Fig. 10, upper panel).

The base case is a 'dike only' system, with stationary, non-vegetated foreshore at 0 m MSL. SLR in the period 1996–2100 leads to an increase in failure probability by a factor of 18, from an initial value of $5{\cdot}10^{-4}$ to $1{\cdot}10^{-2}$ in 2100 (Fig. 10, middle panel), which is equivalent to a difference in dike height of more than 1 m (Table 1). The effect is larger than SLR only, because higher waves can reach the dike at a greater depth. The addition of a vegetated foreshore at 1.5 m MSL (strategy 2, section 2.3) leads initially to a reduction in P_f by a factor of 2.8 (Fig. 10, lower panel), compared to the 'dike only' system. Without morphological adjustment to SLR, this effect decreases to a factor of 2.1 in 2100. Contrastingly, an increasing effect is found in case of accretion equal to SLR (from 2.8 in 1996 to 3.8 in 2100). More realistic accretion, according to Eq. (2), results in a more or less constant factor of 2.8 in P_f. After 50 years (a typical lifetime for structures like seawalls), the failure probability of the 'dike only' system has increased by a factor of 2.7 (Fig. 10, middle panel, black dashed lines). The same increase is reached 4 years earlier in the case of a

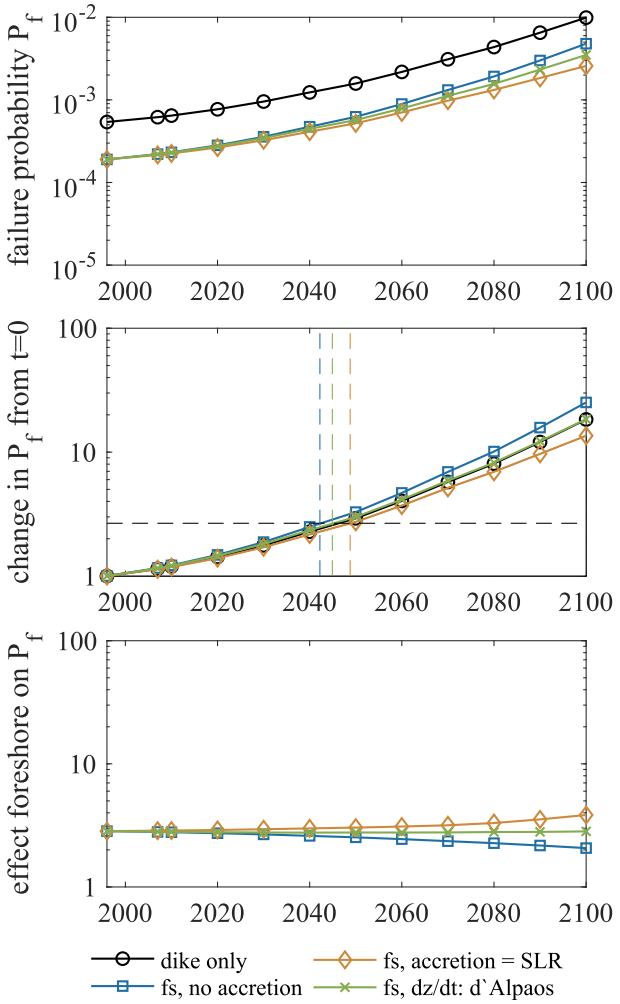

Fig. 10. Top panel: annual failure probability P_f in time for different accretion rates of the foreshore (abbreviation 'fs'). Middle panel: the change in P_f with respect to $t = 0$ due to the combination of SLR and accretion (multiplication factor). Bottom panel: reduction factor of P_f with respect to the 'dike only' system.

foreshore without accretion, and 3 years later with accretion equal to SLR. Therefore, foreshore adjustment to SLR leads to a 7 year difference in lifetime (14%) for this system.

4.2. Lateral marsh dynamics

At the case study site in the Wadden Sea, lateral marsh dynamics are small due to the presence of brushwood dams. Therefore, temporal variations in marsh width B_{fs} are investigated for unprotected salt marshes in the Western Scheldt, an estuary in the South-West of the Netherlands. This analysis is included in Appendix A. Fig. 11 shows, as an example, the result for the salt marsh with the largest variation in width.

Fig. 12 shows how such variations in foreshore width would affect the failure probability of the case study system, with a foreshore at 1.5 m MSL (Fig. 4). Most marshes in the Western Scheldt displayed a limited variation in the considered period of 65 year, see Appendix A. The maximum difference in marsh width of 170 m (Table 4) leads to a change by a factor of 1.34 with respect to the failure probability at a marsh width of 300 m (Fig. 12 and 385 to 215 m). This is equivalent to a difference in dike crest level of 9 cm and a difference in foreshore elevation of 38 cm. As marsh width increases, variations in width have

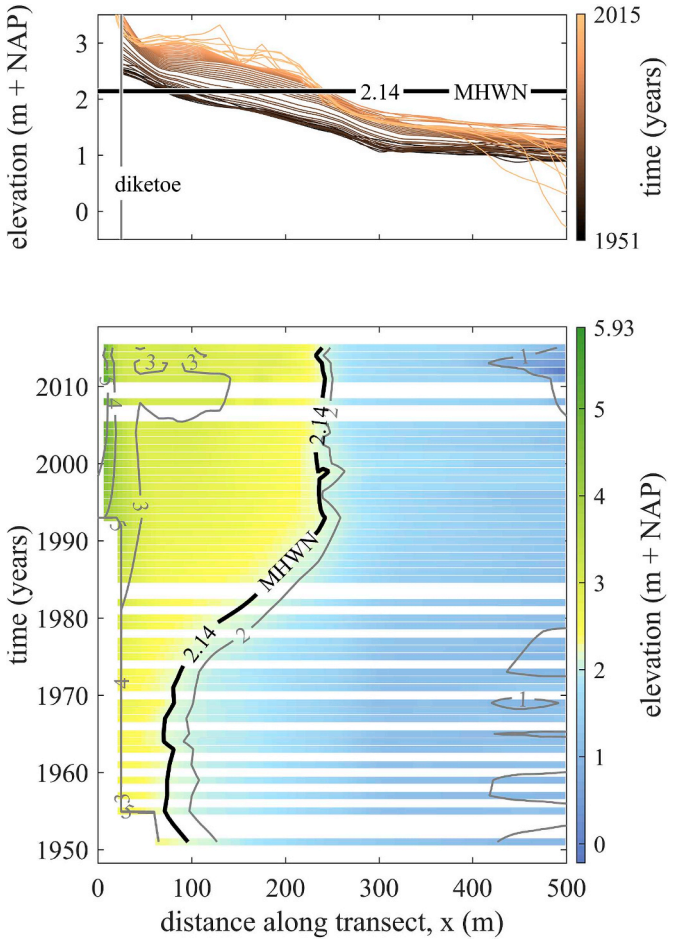

Fig. 11. Profiles of the salt marsh and mudflat at 'Zimmermanpolder' between 1951 and 2001 (top panel), including the vertical position of the marsh edge at MHWN (2.14 m + NAP) and the fixed dike toe position. Old profiles are indicated with darker colors, and recent profiles with light colors. Lateral marsh edge dynamics display marsh expansion (bottom panel). Colors indicate elevation. The higher landward side of the profiles is at $x = 0$, while the lower seaward side is at $x = 500$ m. Gray contour lines are added for clarity; the black line indicates the marsh edge at MHWN. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

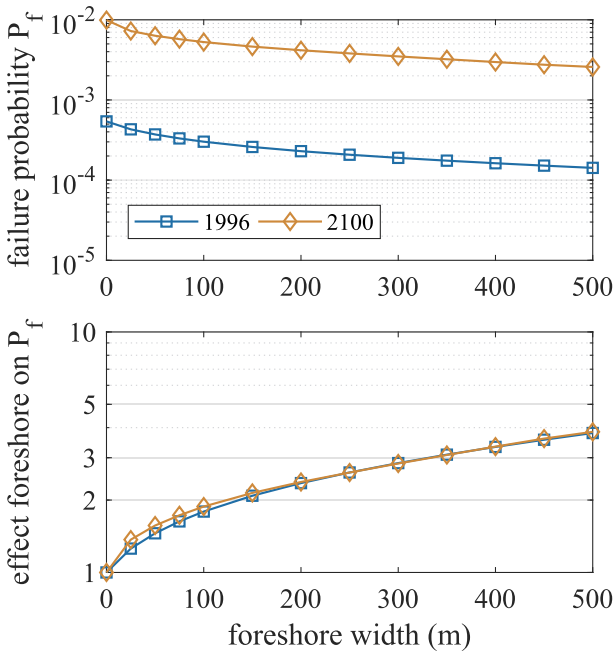

Fig. 12. Variation with foreshore width of (1) the failure probability in 1996 and (after sea level rise) in 2100 (top panel), and (2) the factor with which P_f decreases with respect to a 'dike only' system (bottom panel).

decreasing influence on failure probabilities. The influence of foreshore width does not change noticeably after SLR and subsequent accretion is considered according to Eq. (2) (lower panel).

4.3. Performance of strategies for flood risk reduction

Different strategies for flood risk reduction are considered, as introduced in section 2.3. These strategies are compared in terms of failure probabilities in the period 1996–2100. Effects are expressed in terms of a reduction factor on the failure probability of the 'dike only' system (Fig. 13, lower panel), with crest level at 6 m MSL and foreshore at 0 m MSL.

0. In case of the 'do nothing' strategy, the 'dike only' system with crest level at 6 m MSL displays an increase in failure probability by a factor of 18 in the period 1996–2100 (mentioned in section 4.1 and visible in Fig. 13, middle panel).

1. Dike heightening from 6 to 7 m MSL initially has an effect of a factor of 31, decreasing to 14 in 2100 (Fig. 13, lower panel). This decrease is primarily caused by SLR, and secondly due to subsequent higher exposure to waves.
2. Construction of a 300 m wide foreshore at 1.5 m MSL leads to a reduction of the system's failure probability by a factor of 2.8, with respect to a dike only (Fig. 13, lower panel). This effect is rather constant in the period 1996–2100 due to sediment accretion.
3. Addition of an earthen or rubble mound breakwater at the marsh edge of the same foreshore initially has a relatively large effect: a reduction factor of 5.6, which is 2.0 times larger than the effect of a foreshore only (2.). However, going to 2100, the breakwater's additional effect declines due to increasing submergence. This leads to a decrease in the added value of the breakwater in 2100: only a factor of 1.4 compared to the situation with a constructed foreshore (where a factor of 1.0 means no effect).
4. Further, we consider the foreshore that consists of a high zone directly in front of the dike and a low zone with pioneer vegetation. The initial effect of the foreshore is identical to that of 1 m dike heightening (a factor of 30 in 1996), since the foreshore elevation was iteratively chosen to obtain this failure probability. The repetitive

Table 4
Characteristics of the marsh width (mean, minimum, maximum, largest difference and standard deviation) of multiple marshes along the Northern (N) and Southern (S) shore of the Western Scheldt estuary in the Netherlands. The table includes the assumed vertical position of the marsh edge (m + NAP, where NAP is the Dutch ordinance level, close to MSL), the assessed period and available years of data (vaklodingen) within this period.

Salt marsh	Mean (m)	Minimum (m)	Maximum (m)	Difference (m)	Standard deviation (m)	MHWN (m + MSL)	Period assessed (years)	Available years
Zuidgors (N)	255	230	280	50	13	1.85	1955–2015	37
Baarland (N)	109	95	135	40	12	1.83	1958–2015	36
Biezelingsche Ham (N)	62	50	80	30	7	1.80	1957–2015	46
Zimmermanpolder (N)	148	50	220	170	71	2.14	1951–2015	47
Paulina (S)	246	215	320	105	31	1.73	1955–2015	37
Thomaespolder (S)	90	70	150	80	16	1.71	1955–2015	37
Hoofdplaat (S)	139	130	155	25	8	1.59	1999–2015	13
Hellegat (S)	–	–	–	–	–	1.81	–	–
Average (N)	144	106	179	73	26	1.91	–	42
Average (S)	158	138	208	70	18	1.71	–	29
Average (overall)	150	120	191	71	23	1.81	–	36

earthmoving from the natural marsh to the high zone leads to a reduction factor of 47 in 2100, significantly higher than the factor of 14 of 1 m dike heightening at the same time (Fig. 13, lower panel).

5. Application of brushwood dams results in accretion and salt marsh establishment. This intervention has initially no effect on the system's failure probability. However, in 2100, its effect approaches that of a system with a constructed foreshore (factors of 2.7 and 2.8, respectively, see Fig. 13, lower panel), provided that the dams are continuously maintained and heightened. In 2034, vegetation establishes, and the effect of roughness is added to the wave calculations. In this way, the lifetime of the flood defense can be elongated by 28 years, if 50 years is considered as a reference situation (dashed vertical lines in middle panel of Fig. 13).

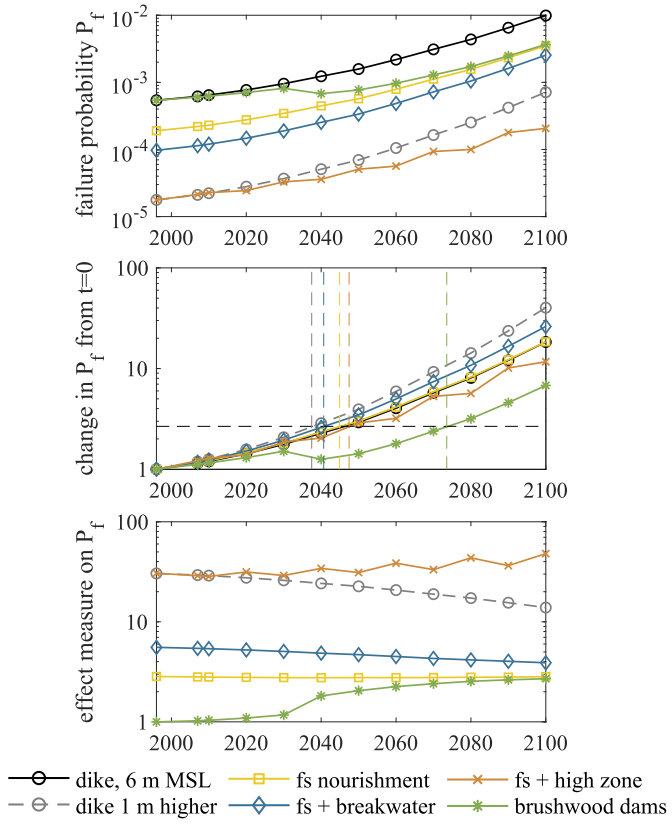

Fig. 13. Top panel: annual failure probability P_f in time for different strategies, including dike heightening and human interventions on the foreshore (abbreviation: fs). Middle panel: the change in P_f with respect to $t = 0$ (multiplication factor). Bottom panel: reduction factor of P_f with respect to the 'dike only' system.

4.4. Cost-effectiveness

In comparing dike heightening and foreshore construction in terms of cost-effectiveness, we first consider 1 m dike heightening and the foreshore with high zone. The high zone initially has an identical effect on the system's failure probability as 1 m dike heightening. This enables straightforward cost-effectiveness comparison of two measures that induce an instantaneous decrease in flood risk. A 1 m dike heightening costs 5.4–14.9 M€, while construction of a 300 m wide foreshore with 100 m wide high zone costs 1.5–3.9 M€. This shows that a high foreshore is a cheaper strategy for coastal protection than dike heightening in this case study, provided that sufficient space and relatively shallow water is present in front of the dike.

Further, we investigate which strategy is most cost-effective, taking into account costs for construction, maintenance, and expected damage after dike failure. Fig. 14 shows for different strategies how the NPV of total costs (annual investments plus annual expected damage) depends on the damage after dike failure.

Without damage ($D = 0$), the 'do nothing' strategy obviously has the minimum total costs. For damage above 70 M€, construction of a foreshore with high zone is more cost-effective. Constructing a foreshore with natural elevation at 1.5 m MSL is more cost-effective than

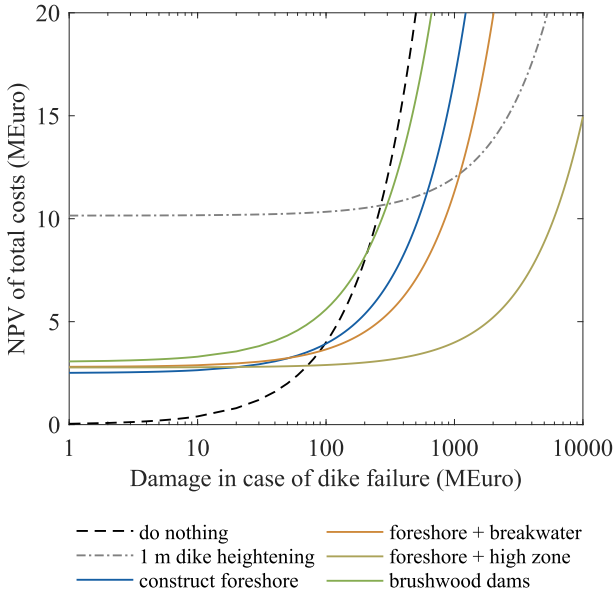

Fig. 14. Total costs (in million €, vertical axis) as a function of damage in case of dike failure (in million €, horizontal axis), for different strategies (different lines). The most cost-effective strategy for a given damage is that with the lowest total costs, which is the NPV of all investments and expected damage.

	Dike heightening	Foreshore construction	Foreshore + breakwater	Foreshore + high zone	Brushwood dams
Cost-effective flood risk reduction	✔	✔	✔	✔✔	o
Required maintenance	o	o	o	✘	✘✘
Initial ecological impact	o	✘	✘	✘	o
Additional eco-system services	o	✔✔	✔	✔	✔✔

Fig. 15. Qualitative assessment of arguments for choosing different strategies for flood risk reduction (checkmark = positive, cross = negative, zero = neutral).

1 m dike heightening for expected damages between roughly 100 M€ and 600 M€ (without breakwater) and between 90 M€ and 1100 M€ (with breakwater). For higher damage, dike heightening is more appropriate. Although sheltering structures such as brushwood or bamboo dams require low initial investments, costs for maintenance are considerable. Further, their effect on safety is only significant after some decades. This postponed effect on flood risk is penalized in the NPV calculation. Therefore, sheltering structures in the current form and associated costs are not attractive from a purely flood risk perspective.

5. Discussion

5.1. Multi-disciplinary evaluation

This paper has assessed the efficiency of vegetated foreshores, primarily looking from the perspective of engineering, flood risk, and economics. However, these are not the only factors that determine the suitability of solutions in the local situation. For example, in the Wadden Sea, each strategy should be carefully assessed in terms of short-term and long-term effects on the large-scale physical system, including its ecology and associated ecosystem services, and should fit into legal boundaries. For the case of (man-made) salt marshes, an assessment is required of the viability of salt marshes within the physical boundary conditions of the large-scale system, including available space, tidal range, sediment concentration, flow intensity, and wave action. Further, the ecological value of the salt marsh itself should be investigated, as well as interactions between the salt marsh and neighboring aquatic ecosystem.

Traditionally, flood defenses are located within their predefined narrow borders, fully apart from the ecosystem in front. Dike reinforcement and ecological restoration are performed separately by different institutions. Dike heightening can easily be applied within the zone that is designated for flood protection. In contrast, when working with foreshores, measures should fit in the large-scale physical and ecological system, which requires a multi-disciplinary approach to flood protection. Although decision making can be more challenging and time-consuming, combined benefits for flood protection and nature conservation can potentially be improved if such an integrated approach is adopted (Janssen et al., 2015).

Adaptive flood risk management comprises flexibility in measures and strategies, in order to allow for speeding up or slowing down. This way, flood risk managers avoid undesirable exposure to flood risk and exorbitant investments that may be done in response to uncertain future climate change and demographic developments (Klijn et al., 2015). Working with vegetated foreshores fits well into an approach based on adaptive flood risk management for two main reason: the ability of vegetated foreshores to raise with sea level by natural sediment accretion, and ease of incremental upscaling of a foreshore compared to repetitive dike reinforcement.

Important criteria for choosing a certain flood risk reducing strategy are (1) cost-effectiveness for flood risk reduction, (2) required maintenance, (3) initial impact on the local ecosystem during construction, and (4) long-term additional ecosystem services. Here, we qualitatively evaluate the strategies considered in this paper, looking at those four aspects (Fig. 15).

1. Dike heightening and construction of a foreshore with a high zone are considered most positive for cost-effective flood risk reduction. Both strategies have a large effect on flood risk, but dike heightening is more expensive. Construction of a foreshore that resembles a natural salt marsh (either with or without breakwater) has less effect on flood risk, since its elevation is limited to approximately mean high water, where natural accretion can take place. The effect of brushwood dams on flood risk is on the long run similar to that of foreshore construction. However, due to continuous maintenance costs and postponed benefits for safety, it is less attractive in terms of life-cycle costs than the other strategies considered here.
2. Concerning required maintenance: dike maintenance is needed in all strategies, and is neglected in this comparison. In addition, periodic earthmoving is applied for the foreshore with high zone, in order to keep pace with sea level rise. Brushwood dams demand more or less continuous maintenance. Foreshores with gentle slopes are assumed to be morphologically stable here, although this is highly case-specific. At locations where marsh edge erosion is to be expected, periodic sediment nourishment is needed, which seriously increases life-cycle costs and ecological impact. Addition of brushwood dams or other sheltering structures may be considered here for erosion mitigation.
3. Initial ecological impact of dike heightening is neutral, since it does not significantly affect the estuary or coastal sea, either positively or negatively. Foreshore construction and earthmoving have adverse ecological impacts, due to for example disturbance of underwater habitats and increased turbidity due to dredging and nourishment. Impact can be reduced by using sediment with similar grain size as the native bed material (McLachlan, 1996), and by avoiding nourishments in spring, considering the reproduction cycle of many benthic species (Menn et al., 2003). Brushwood dams have negligible adverse environmental effects. These scores are only related to short-term impact.
4. Systems with a high nature-based character provide additional ecosystem services, such as providing habitats for fish and other wildlife, recreation, carbon sequestration, water purification, and erosion control (Barbier et al., 2011). Long-term effects on the surrounding ecosystem, either positive or adverse, should be investigated in a site-specific ecological impact assessment. Here, salt marsh realization via nourishment or brushwood dams are considered most positive on this aspect, since these strategies aim to realize a foreshore that resembles a natural salt marsh, with salt-tolerant vegetation, inundated during high tide. These strategies can be considered as 'Building with Nature', since natural materials and processes are exploited for safety. Artificial elements, such as a breakwater or high zone, decrease the nature-based character of the system. More research is required to quantify ecological performance of different foreshore configurations.

5.2. General applicability

This study demonstrates that the cost-effectiveness of vegetated foreshores depends on three main factors:

1. The original failure probability of the system, and how much this probability can be affected by foreshores, both initially and after sea level rise;
2. The investments required to construct and maintain foreshores, in comparison to hard structures;
3. The economic value of the protected area, where nature-based solutions are relatively more attractive for low economic value.

These three factors should be quantified on a local scale, accompanied by assessments of (1) viability in the surrounding physical system and (2) ecological impact, in order to decide whether nature-based solutions can outcompete traditionally engineered structures.

This paper's integrated analysis of wave load reduction, probability of failure, vertical salt marsh adaptation to sea level rise, and life-cycle costs can be applied to other estuaries and coastal seas worldwide. Examples of interesting future applications in low-lying coastal regions are mangrove coasts in the Mekong delta (Vietnam), or coastal wetlands along the coasts of Virginia and Louisiana (US). Previous studies in these areas have quantified effects on surge and waves (e.g., Bao (2011); Glass et al. (2018); Wamsley et al. (2010)). By applying an approach similar to that in the current study, quantitative insights can be obtained on the cost-effectiveness of such nature-based solutions for flood risk reduction.

5.3. Future work

Starting from the current study, research concerning salt marshes for flood risk reduction can be brought a step further by collecting and describing more practical examples where vegetated foreshores have been implemented, including construction and maintenance costs. We encourage to extend existing studies on cost-effectiveness of nature-based solutions, such as Reguero et al. (2018), with site-specific calculations of flooding probabilities and the influence of nature-based solutions.

In most tables and handbooks, bottom roughness values are prescribed for various vegetation types, without taking into account the disruption of vegetation during storms and hurricanes. Also in this paper, a standard empirical roughness value has been used. Predictions of wave attenuation over wetlands can be improved by calibrating bottom roughness in numerical models for flow and waves, specifically for salt marsh surfaces that have experienced extensive stem breakage.

The D'Alpaos equation for vertical marsh accretion (Eq. (2)) a is rather simple analytical formula, which could easily be used to estimate the response of marshes to sea level rise, tidal amplitude, and sediment concentration. We did not validate its performance. The purpose of this study was to show the importance of a realistic vertical accretion rate for long-term effectiveness of foreshores in reducing flood risk. There was no need to select the best possible model. More sophisticated models such as SLAMM, used in e.g., Craft et al. (2009), could provide more precise estimates in future studies, provided that the simulations do include the feedback mechanisms that allow marshes to adapt to SLR by accelerating rates of elevation change (Kirwan et al., 2016).

This paper puts central the flood risk reduction and associated cost-effectiveness of nature-based solutions, which are the main criteria for choosing a certain flood risk reduction strategy. Nature-based solutions however, typically require joint action of multiple different stakeholders, including nature organizations and local governments. First, because these measures provide more services than flood risk reduction only. And second, because these solutions should often be realized in ecologically sensitive and protected areas. Therefore, successful implementation of nature-based solutions requires different governance and institutional arrangements compared to traditional flood risk management (Janssen et al., 2015; Borsje et al., 2017). More research on this aspect can help in getting nature-based solutions in the mainstream of coastal protection. In addition, the interplay between human interventions in salt marshes and biodiversity needs to be studied, to optimize the whole spectrum of ecosystem services provided by natural and man-made salt marshes.

The economic value of ecosystem services other than flood risk reduction have not been taken into account in the current study. This aspect should be added to the comparison between different strategies, in order to perform a fully integrated cost-benefit analysis. In this study, vegetated foreshores are considered as ecologically valuable components of the large-scale physical system. However, connectivity between the terrestrial high salt marsh and the lower aquatic parts of the system influences certain ecosystem services (Barbier et al., 2011). Therefore, for actual applications of vegetated foreshores, we underline the recommendations made in Bockstael et al. (2000), who stressed that the overall ecosystem services provided by the large-scale system should be investigated, instead of valuing specific and localized components of the small-scale ecosystem only.

6. Conclusions

This paper presented an approach to assess the cost-effectiveness of salt marshes for flood risk reduction by coupled calculations of wave attenuation, sediment accretion under sea level rise, probability of dike failure, and life-cycle costs. Different interventions on the foreshore were compared with traditional dike heightening, considering the net present value of benefits (failure probability reduction) and costs (for construction and maintenance).

Sea level rise leads to increasing probabilities of dike failure by wave overtopping, caused by a combination of rising still water levels and decreasing wave height reduction over the foreshore. Marsh elevation change due to sediment accretion mitigates the increase in wave height, and elongates the lifetime of a dike-foreshore system.

Cost-effectiveness of nature-based solutions versus hard structures depends on various local conditions, most importantly: economic value of the protected area, current failure probability, reduction of this probability via foreshores, and costs for construction and maintenance.

Different strategies were assessed for a case study in the shallow Dutch Wadden Sea in the Netherlands. Salt marshes here are exposed to relatively high day-to-day waves and tidal currents, which leads to stable plant species, firmly rooted in mineral soils. Such marshes are highly resistant to surface erosion, even during extreme storms. The calculations give rise to the following conclusions from an engineering perspective:

- Salt marsh construction is cheaper than dike heightening. However, salt marshes are limited in effect on failure probabilities because of their dependence on sediment accretion in the intertidal zone. For the considered case study, salt marsh construction is only more cost-effective than dike heightening, if small to moderate economic damage occurs in case of dike breaching.
- Artificial high zones and breakwaters on the salt marsh improve the flood defense's reliability substantially, against relatively low costs. A foreshore with high zone is even more cost-effective than dike heightening if it is constructed well above mean high water. Without human interventions, breakwaters and high zones lose effect because of SLR and absence of natural sediment accretion. However, periodic earthmoving from the pioneer zone to the high zone is an effective alternative for obtaining persistent flood risk reduction.
- Sheltering structures such as brushwood or bamboo dams enhance sediment accretion and lead on the long term to similar foreshore effects as instant construction via nourishment. However, continuous maintenance costs and postponed benefits for safety discard brushwood dams as attractive strategy for flood risk reduction.

These conclusions are valid for the considered case study and design options. We emphasize that similar site-specific analyses are required, before results can be universally applied to other systems and locations.

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A. Temporal variations in Marsh Width

This appendix contains an analysis of temporal variations in marsh width, based on historical bathymetric survey data. At the case study site in the Wadden Sea, lateral marsh dynamics are small due to the system of brushwood dams. Therefore, we investigate temporal variations in marsh width B_{fs} for unprotected salt marshes in the Western Scheldt, an estuary in the South-West of the Netherlands. For these marshes, a lot of data is available, in contrast to the Wadden Sea, where only a limited number of bathymetric surveys have an extent that covers the salt marshes. The selection of study sites in the Western Scheldt is identical to that of Van der Wal et al. (2008).

A.1. Methods

Lateral dynamics of salt marshes in the Netherlands are captured in an extensive bathymetrical dataset (so-called 'Vaklodingen') of the Dutch nearshore areas. These bathymetric data are collected since 1925–1935 by the Ministry of Infrastructure and the Environment (former Ministry of Transport, Public Works and Water Management), interpolated over a 20 × 20 m grid (De Kruif, 2001; Wiegman et al., 2005). Temporal dynamics of the salt marsh and tidal flats are obtained by linear interpolation of the Vaklodingen data to a transect perpendicular to the marsh edge. For determining the marsh width, the (most recent) position of the dike toe is used as the landward boundary. Since salt marsh area was not collected for the full period, the marsh edge position is determined by using a tidal benchmark. Different tidal benchmarks have been used in the literature for defining marsh edge positions (Balke et al., 2016), e.g. 20–40 cm below Mean High Water (MHW) in the Dutch Wadden sea (Bakker et al., 2002), or Mean High Water Neap (MHWN) in the Western Scheldt Van der Wal et al. (2008). In this paper, MHWN is used. Finally, the marsh width is defined as the distance from the dike toe (landward reference) to the marsh edge (seaward reference). Missing data at the intertidal area were interpolated in time. To prevent extrapolation above MHWN, measurements obtained in years before the first measurement reached up to MHWN, were removed.

A.2. Results

The width of the analyzed marshes varied over the assessed period of 65 years (Table 4). The largest variation was observed at Zimmermanpolder, with a marsh width increasing from 50 m in 1951 to 220 m in 2015 (Fig. 11). The marsh width of most marshes remained constant or increased in time, except of Paulina, where net erosion was found. In general, the marsh width of the salt marshes located at the northern shore of the estuary were smaller than the marshes at the southern shore, although the standard deviation in marsh width at the northern shore was on average larger. These findings are in line with Callaghan et al. (2010) and Van der Wal et al. (2008). For Hellegat, no results were obtained, since the profiles did not exceed MHWN.

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